LCA Methodology

Evaluating Alternative Life-Cycle Strategies for Electrical Appliances by the Waste Input-Output Model

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Abstract

Goal, Scope and Background. In 2001, a new law on the recycling of end-of-life electric home appliances (EL-EHA) was put into effect in Japan; it was the first legislation of its sort in the world, and deserves to be called the 'Japan model.' This article is concerned with the LCA of alternative life-cycle strategies for EL-EHA, which consist of recycling as prescribed by the law, 'ecodesign' strategies such as the implementation of design for disassembly (DfD) and the extension of product life (EPL), with and without ex-post functional upgradability, and the once-dominant treatment methods such as landfilling and simple shredding.

Methods. We use the waste input-output (WIO) analysis, a new method of hybrid LCA that was developed by the authors [1]. The WIO extends the conventional input-output analysis to explicitly take into account the interdependence between the flow of goods and the flow of waste in the whole economy, and hence provides an optimal platform for LCA involving waste treatment and recycling. Furthermore, the WIO enables us to evaluate not only environmental impacts, but also economic impacts such as sectoral output and employment. Our analysis is based on the WIO table for 1995 and detailed process data on recycling.

Results and Discussion. Recycling was found to outperform the 'traditional' treatment method strategy with regard to the reduction of CO_2 emission, landfill consumption, and the demand for abiotic resources. Thanks to efficient utilization of the existing retail network system, it was also found to be more efficient economically. Additional implementation of DfD and the associated increase in the quality of recovered materials (plastics) were found to augment the positive environmental impacts of recycling. The EPL without upgrading resulted in a significant reduction in the environmental impacts, but also in the level of employment. On the other hand, the EPL with upgrading was found to outperform the recycling strategy in terms of environmental impacts without having significant negative economic impacts.

Conclusion and Recommendation. Recycling of EL-EHA, as prescribed by the Japanese law on the recycling of EL-EHA, was found to be effective in reducing CO_2 emission, depletion of abiotic resources, generation of waste, and landfill consumption, provided the rate of retrieval remains at a high level. Our results also indicate the possible effectiveness of ecodesign strategy toward the realization of a sustainable economy.

Keywords: Design for disassembly; ecodesign; electrical appliances; product life; recycling; waste input-output analysis

Introduction

In Japan, the number of discarded TV sets, refrigerators, washing machines, and air conditioners amounts to about 2 million units, or 0.73 million tons, annually. Direct landfilling or landfilling after shredding, with iron recovery, used to be the main treatment method applied for them. With the aim of increasing the recovery of waste materials and reducing the negative environmental impact associated with such treatment, the Japanese government introduced a new law on the recycling of End of Life Electric Home Appliances (EL-EHA) in April 2001. The law makes the manufacturers of electrical home appliances (EHA), TV sets, refrigerators, washing machines, and air conditioners, responsible for the recycling (re-commercializing, to be precise) of specific percentages of the respective EL-EHA, and makes the consumers (disposers) responsible for bearing the cost of recycling and transport. This was the first legislation of its sort in the world, and deserves to be called the 'Japan model.'

Characteristic of the Japan model is the use of an intensive material recovery process that makes recovery of material components from EL-EHA possible at substantially higher rates than the conventional shredding processes [2]. With regard to the use pattern of recovered materials, however, the Japan model is still largely characterized by 'downcycling' [3] because the original value of the product is destroyed and only some natural resources are recovered.

It is often pointed out that the current manufacturing system represented by the combination of mass production, mass consumption and mass downcycling will not be a sustainable one [4]. The current system is characterized by a loop of product life cycle that is large and open. It is argued that the loop should be transformed to a small and closed one in order to make the manufacturing system sustainable [5]. Maintenance and upgrading of products to extend their lifetime, and the reuse of disused components, among others, emerge as important strategies for achieving this transformation.

This article is concerned with the evaluation of the environmental and economic effects of these alternative life-cycle strategies by use of the waste input-output (WIO) analysis [1]. A combined use of input-output analysis (IOA) and the traditional process-based method has become a standard tool of LCA owing primarily to the public availability of data, ease of computation, and the well-defined system boundary [6,7]. IOA, however, is not appropriate to deal

with issues related to waste management, because it does not consider the flow of waste. The WIO is an extended form of IOA that takes into account the interdependence between the flow of both goods (and services) and waste. Using the WIO, we can evaluate the impact of alternative product life-cycle strategies on emissions such as CO_2, landfill consumption, and economic activity such as sectoral output and employment.

Associated with EL-EHA recycling is a complicated flow of goods, recovered waste materials, and residues among a large number of different sectors of the economy, which consist of the EL-EHA treatment sector, the users of recovered materials, and the treatment sector of residues. Furthermore, EL-EHA can be treated in several different ways, each of which is characterized by different combinations and types of inputs, recovered resources and residues. For an LCA of EL-EHA, it is thus of great importance to properly take into account this interdependence between the flow of goods and waste over different sectors of the economy. The advantage of WIO-LCA consists in that it is based on a consistent accounting system (WIO table) that describes this interdependence.

Prior to the introduction of the recycling law in 2001, there was a worry about a possible increase in the illegal dumping of EL-EHA. Japanese experiences of the past years, however, indicate that this worry did not materialize [8]. Furthermore, with the enforcement of the recycling law, the once sizable export of EL-EHA to Asian, and South and Central American, countries has been declining. The ban of the import of EL-EHA in the form of products in China since 2000 has also worked to strengthen this declining tendency [9]. In this paper, we assume that all the EL-EHA are retrieved, and focus attention on the effects that result from the application of alternative treatment processes.

The article is structured as follows. First, we give a brief summary of WIO in Section 1. Section 2 then describes the setup of alternative life-cycle scenarios, and the associated material recovery and use patterns. Empirical results of our WIO analysis are presented in Section 3. Discussion of its implications and the limitations of analysis close the article.

1 LCA by WIO

Let there be n^I goods- and service-producing sectors (henceforth 'goods sector'), n^{II} waste treatment sectors, and n^w waste types. For ease of exposition, we define the sets of natural numbers referring to each of these sectors and waste types as $N^I := \{1,\dots,n^I\}$, $N^{II} := \{n^I+1,\dots,n^I+n^{II}\}$, $N := N^I \cup N^{II}$, and $N^w := \{1,\dots,n^w\}$. Let x_I be an $n^I \times 1$ vector, the i-th component of which is the output of goods sector $i \in N^I$, and x_{II} be an $n^{II} \times 1$ vector, the $(i - n^I)$-th component of which is the output of waste treatment sector $i \in N^{II}$. We measure the output of a waste treatment sector by the amount of waste it treated. We denote the $n^I \times 1$ vector of the final demand for goods as $X_{I,F}$, and the $n^w \times 1$ vector of the generation of waste from the final demand sector $W_{\bullet,F}$.

Let $A_{I,II}$ be an $n^I \times n^{II}$ matrix of input coefficients, the (i,j)-component of which refers to the input of good $i \in N^I$ per unit of output of $j \in N^{II}$, and $G_{\bullet,I}$ be an $n^w \times n^I$ matrix of net waste generation coefficients, the (k,j)-component of which refers to the net generation (generation net of recycling) of waste $k \in N^w$ per unit of output of sector $j \in N^I$. $A_{I,I}$ and $G_{\bullet,II}$ are defined in a similar manner. Finally, let S be an $n^{II} \times n^w$ non-negative matrix whose (j,k)-component s_{jk} represents the share of waste $k \in N^w$ that is treated by treatment method $j \in N^{II}$.

The WIO then gives a simple representation of the relationship between the level of industrial output x_I and waste treatment x_{II}, the structure of technology represented by $A_{I,I}$, $A_{I,II}$, $G_{\bullet,I}$, and $G_{\bullet,II}$, and the lifestyle of final consumers represented by $X_{I,F}$ and $W_{\bullet,F}$ [1]:

$$\begin{bmatrix} A_{I,I} & A_{I,II} \\ SG_{\cdot,I} & SG_{\cdot,II} \end{bmatrix} \begin{bmatrix} x_I \\ x_{II} \end{bmatrix} + \begin{bmatrix} X_{I,F} \\ SW_{\cdot,F} \end{bmatrix} = \begin{bmatrix} x_I \\ x_{II} \end{bmatrix}, \tag{1}$$

or

$$\begin{bmatrix} x_I \\ x_{II} \end{bmatrix} = \left(I_n - \begin{bmatrix} A_{I,I} & A_{I,II} \\ SG_{\cdot,I} & SG_{\cdot,II} \end{bmatrix} \right)^{-1} \begin{bmatrix} X_{I,F} \\ SW_{\cdot,F} \end{bmatrix}, \tag{2}$$

where I_n is the identity matrix of order $n := n^I + n^{II}$. Let R be an $n^E \times n$ matrix of the emission coefficients of environment loading factors with n^E referring to the number of loading factors. The emission of loading factors induced by the lifestyle $(X_{I,F}, W_{\bullet,F})$ is then given by

$$R\begin{bmatrix} x_I \\ x_{II} \end{bmatrix} = R\left(I_n - \begin{bmatrix} A_{I,I} & A_{I,II} \\ SG_{\cdot,I} & SG_{\cdot,II} \end{bmatrix} \right)^{-1} \begin{bmatrix} X_{I,F} \\ SW_{\cdot,F} \end{bmatrix}. \tag{3}$$

In (3), the introduction of a new treatment and/or recycling technology occurs as a change in the coefficient matrices A, G, R (for simplicity, we ignore the suffixes 'I' and 'II'), and a change in lifestyle occurs as a change in final demand vectors $X_{I,F}$, $W_{\bullet,F}$. For instance, let Δa_{ij}, Δg_{ij} and Δr_{ij} be the incremental changes in input, waste generation, and emission coefficients associated with the introduction of a certain scenario. The new set of corresponding input, waste generation, and emission coefficient matrices A', G' and R' are then given as $A' = [a_{ij} + \Delta a_{ij}]$, $G' = [g_{ij} + \Delta g_{ij}]$ and $R' = [r_{ij} + \Delta r_{ij}]$. We can evaluate the impact associated with the scenario by comparing the new solution for (3) based on A', G' and R' with the reference solution based on the coefficients before the change.

In this paper, we used the Japanese WIO table for 1995 [10,11]. The WIO table comprises eighty industry sectors, five basic treatment methods (composting, gasification, shredding, incineration, and landfilling), and thirty-six waste types, including nine types of bulky waste. Incineration was further broken down into several types, depending on the size of incinerator, methods of energy recovery, and the treatment of incineration residues. While the WIO model can deal with an arbitrary number of emissions, only CO_2 is considered in the following due to data constraints. Because the WIO table is based on the Japanese IO table, its system is bounded by the territory of Japan. The possible environmental and economic effects associated with the activity of exporting countries are not considered.

2 Life-Cycle Scenarios

2.1 Recovery and use of waste materials

Table 1 shows the amount of EL-EHA and their representative material composition. We first consider four scenarios with regard to the recovery and use of material components. They consist of landfilling (**Lf**), shredding (**Sr**), recycling (**Rc**), and recycling with Design for Disassembly (**DfD**), to the explanation of which we now turn.

Landfilling (Lf): EL-EHA are directly landfilled without any pretreatment, except for recovery at a recovery rate of 90% and decomposition of chlorofluorocarbon (CFC) 12. In all the scenarios we consider, CFC12 is treated this way.

Shredding (Sr): EL-EHA are shredded and their iron component is recovered at a recovery rate of 99%. The shredding process consists of a set of crushers and magnetic separators. Recovered iron scraps are used as material for steel making in electric arc furnaces. Steel from an electric arc furnace is a substitute for converter steel that uses iron produced by a blast furnace. The remaining un-recovered components become shredder dust and are landfilled.

Recycling (Rc): EL-EHA are subjected to an intensive material recovery process that is characterized by the integrated use of sorting and initial disassembly, cryogenic crushing, low temperature shredding and metallic-plastic compound separation, and copper-aluminum separation, as well as PCB (printed circuit board) solder recovery [2]. Aside from iron (at a recovery rate of 99.9%), other materials such as plastics (at a recovery rate of 49%), glass (at a recovery rate of 90%), copper (at a recovery rate of 90%) and aluminum (at a recovery rate of 92%) are also recovered. Furthermore, CFC11 contained in the urethane foam of refrigerators is also recovered (at a recovery rate of 90%) and decomposed.

Recovered copper, aluminum, and glass are used respectively for copper elongation, aluminum rolling, and glass making as substitutes for virgin materials. Plastics are used as a reductant in place of coke in blast furnaces of the iron and steel industry. Due to the small amount of solder recovered from PCBs (about 1% of the weight of PCBs [2]) and the lack of detailed information on its recycling, we put together PCBs with 'other plastics' and ignored the recycling of nonferrous metals other than copper and aluminum. For similar reasons, we do not consider the disposal of residues generated in the decomposition process of CFCs.

This scenario satisfies the rate of material recovery prescribed by the Japanese EL-EHA recycling law by a wide margin (to be precise, the law prescribes re-commercialization ratios of around 50 to 60%, the satisfaction of which is more stringent than recycling ratios because the recovered materials have to be sold at positive prices).

Recycling with DfD (DfD): In order to take into account the fact that "the environmental impacts of products are determined to a high percentage in the design phase" [12], and that disassembling is "the first and most important point in the recycling process" [13], we consider the case where the implementation of DfD significantly improves the efficiency of disassembling and the quality of recovered materials as well. In particular, we consider the case in which the efficiency of the disassembling process is increased by 50%: all the input coefficients related to the shredding activity are halved (improvement to this order of magnitude is reported for a conceptual model of washing machine [14]). As for the improvement in the quality of recovered materials, it is assumed that 80% of polypropylene (PP) and polystyrene (PS) is recovered at a level of purity that allows them to be used as materials for industrial plastic products, while the remaining plastics are 'downcycled' as a reductant in blast furnaces. Implementation of DfD will also change the manufacturing process of EHA, and hence affect the corresponding input and waste generation coefficients. Owing to a lack of relevant information, this point is subject to a high degree of uncertainty, which needs to be dealt with by a sensitivity analysis.

Table 1: Discard and Material Composition of Electric Home Appliances

	TV set	Refrigerator	Washing machine	Air conditioner
Discarded appliances				
Units (10^3)	9031	4071	4530	3023
Weight (10^3 tons)	225.8	240.2	113.3	154.2
Material composition (%)				
Iron	9.7	49.0	55.7	45.9
Copper	1.5	3.4	2.9	18.5
Aluminum	0.3	1.1	1.4	8.6
Other metals	1.4	1.1	0.5	1.5
Plastics: PP & PS[a]	15.0	22.1	28.7	9.3
Plastics: PVC[a]	0.5	3.4	2.0	1.9
Plastics: others	0.5	17.8	4.0	6.4
Glass	62.4	0.0	0.0	0.0
PCB[a]	8.1	0.0	1.5	3.1
CFC11		0.77		
CFC12		0.33		2.0
Others	0.5	1.0	3.3	2.9

Note: Units in parentheses
[a] PP, PS, PVC, and PCB stand for polypropylene, polystyrene, polyvinyl chloride, and printed circuit boards, respectively
Source: [2,21]

2.2 Collection and transport

Significant differences exist between the conventional scenarios (**Lf** and **Sr**) and **Rc** with regard to the method of collection and transport distance, which need to be taken into account in our analysis.

First, different subjects are involved in the collection of EL-EHA from individual households. This brings about significant differences in the collection methods. Under **Lf** and **Sr**, EL-EHA are collected by municipalities based on the curbside collection system, whereas they are collected by retailers of EHA under **Rc**. The utilization of existing retail networks in the collection of EL-EHA is an important feature of the Japanese EL-EHA recycling system, which is made possible by the fact that the generation of EL-EHA usually takes place simultaneously with a new purchase (delivery) of EHA. The same vehicle that delivered the new EHA can then be used to collect and transport the EL-EHA to the nearest, designated collection depots that are operated by EHA manufacturers (there are about 380 such depots across the country). It was assumed that no extra inputs are required for the collection of EL-EHA under **Rc**. Our specification of the curbside collection system follows [15], where a 4-ton truck manned with three persons runs at a speed of 10 km/h collecting waste at each curbside.

While **Rc** may outperform **Lf** and **Sr** in terms of the efficiency of collection, it may not prove to be effective in terms of the transport of collected EL-EHA to recycling facilities. This is so because the number of recycling facilities is much smaller (there are forty facilities across the country as of October 2002) than landfill sites or conventional shredding facilities; the collected EL-EHA then have to be transported over substantially longer distances. It was assumed that, under **Lf** and **Sr**, collected EL-EHA need to be transported over a one-way distance of 12 km (this value was taken from [15]), whereas under **Rc** they have to be transported over 75 km. We assume that the transport under **Rc** is done by a 10-ton truck manned with one person, the average speed of which is 40 km/h. Given the hypothetical nature of our specifications of transport distance, a sensitivity analysis will be performed below.

2.3 Extension of product life

The scenarios introduced above are concerned with the recovery and use of EL-EHA components under a given use pattern of EHA on the side of the consumer. An important factor of the use pattern on the side of the consumer is the length of product life. The life of a product has two dimensions: physical life and functional (value) life. It is widely observed, at least in advanced economies, that many products are discarded at the end of their value life, although their physical life still remains, because the consumers find the products they own to be functionally outdated and the possibility of ex-post (after purchase) functional upgrading is zero for most products.

It then follows that a simple extension of physical life will not be sufficient for extending product life; it will also be necessary for products to be designed to accommodate functional upgradability [4]. Below, we consider two scenarios on the extension of product life, which differ from each other with regard to the possibility of functional upgradability.

Extended life with patience (ExP): The first scenario corresponds to the case where the consumers simply use the products (EHA) longer (within the range of the physical lives) even though they may be functionally outdated. To be specific, we consider the case where the life of EHA is extended by 50%. Assuming the stationary situation, where both the purchase and discard of EHAs change simultaneously, the new purchase of EHA and the disposal of EL-EHA are reduced by 33.33% (=1–1/1.5), while the expenditure for repair is increased by 50% (this point will be subjected to a sensitivity analysis to take into account the *likely* case where failure rate increases according to time).

In 1995, Japanese households owned about 271 million units of four types of EHA, spent 5,610 billion yen on the purchase of new EHA, and about 80 billion yen on their repair [16,17], giving annual repair expenditure of about 300 yen per unit of EHA. Because the expenditure for repair is negligibly small compared with the reduction in the expenditure for the purchase of new appliances, the total consumer expenditure is reduced by 0.48%.

We use the term 'extended life with patience', or **ExP**, to represent this scenario because it requires patience on the side of the consumers in the sense that they are asked to use the products over longer periods, even though the products are functionally outdated.

Extended life with functional upgrading (ExU): The second scenario corresponds to the case where the extension of product life is accompanied by ex-post functional upgrading [4]. With regard to the extent to which product life is extended, this scenario is the same as **ExP**. The consumers, however, are no longer asked for patience, because the products in their possession are steadily updated functionally. The way this strategy can actually be implemented may greatly differ for individual products and their specifications, and it is difficult to conceive of an omnibus relationship between repair and update expenditure, on the one hand, and an extension of functional life, on the other.

For the sake of computation, a simple hypothetical case was considered where the same amount of expenditure for a new purchase that was saved by the extension of product life needs to be spent on repair and update to keep it functionally updated. This results in a 2400% increase in expenditure for repair and maintenance on the side of the consumer. Economically, at least, this scenario is of interest because the total consumer expenditure remains unchanged, whereas it is reduced under **ExP** because there is no corresponding increase in the expenditure for repairs that compensates for the decline in the expenditure for new purchases. We use the term 'extended life with functional upgrading', or **ExU**, for this scenario.

Under **ExP**, the total consumer expenditure, the largest component of GDP, is reduced by 0.48%, whereas it remains at

the same level under **ExU**, although its composition changes. It is important to note that repair is not free from waste generation; according to the Japanese WIO table, repair to the amount of one million yen generates, among other things, 7 kg of waste paper, 1 kg of waste plastics and 150 g of iron scrap [11]: under **ExU** the increase in repair is counteractive to the reduction of waste. As for the recovery and use of waste materials from EL-EHA, **Rc** is applied to both **ExP** and **ExU**.

Before leaving scenario set-ups, an important remark is due on the limitation of our analysis. While we are concerned with an LCA of durables, we do not consider issues of dynamic adjustments where EHA of different 'vintages,' with possibly different energy efficiency and material components, become obsolete and are replaced by newer ones. The case we consider corresponds to a sort of stationary state where, within the range of 'extended lives,' EHA of different vintages, including the newest ones on the market, are exactly the same. The lifecycle benefits that result from innovation in product design are limited to only the extension in the functional life of EHAs. The issues related to the timing of the replacement of older products with newer ones, which possibly embody more efficient technology [18], are beyond the scope of this article.

2.4 Inventories

Table 2 shows the inventory of the three recovery processes with regard to major input items, their unit prices, and the WIO industry sectors to which these input items correspond. Panel **A** of **Table 3** then shows the major input and emission coefficients for shredding, recycling, and recycling with DfD. Note, for the first four input items (from water supply to chemicals), that the values were obtained by multiplying the physical quantities by the corresponding prices in Table 2.

The input of general machinery refers to the annualized value of construction costs (1.1 billion yen for a shredding plant and 2 billion yen for an integrated recycling plant with an annual capacity of 18,000 t = 500,000 units × 36 kg per unit) based on the assumption that the plant lasts for 20 years and the general machinery is the sole supplier of the plant facilities.

The inputs of petroleum products and car maintenance refer to the collection and transportation of EL-EHA described in Section 2. Employment refers to total labor requirements that include both vehicle drivers and plant operators/workers. Panel **B** below shows the amount of shredder dust that is generated per ton of treatment under each of the three recovery processes. Because plastics constitute the largest component of shredder dust, it is no wonder that its generation is the smallest under **DfD** and is the largest under **Sh**.

Table 2: Inventories of Alternative Recovery Methods

	Unit	Input per ton of EL-EHA			Price (yen/unit)	WIO classification
		(Lf) Landfilling	(Sr) Shredding	(Rc) Recycling		
Water[a]	L	2.2	2.2	38.8	3.0	Water supply
Electricity	kWh	5.9	109.1	135.6	21.0	Electric power
Slaked lime[a]	kg	0.1	0.1	0.3	20.0	Misc. stone and clay products
Liquid nitrogen[b]	kg			85.4	25.0	Chemical industry

Note: See [11] for the WIO classification of sectors
[a]Used for CFC degradation; [b]Used for low temperature shredding
Source: [2]

Table 3: Coefficients of Recovery Processes

Scenarios	(Sh) Shredding	(Rc) Recycling	(DfD) Recycling with DfD
A. Major input and emission coefficients[a]			
Water supply	0.0007	0.0116	0.0058
Electric power	2.2909	2.8453	1.4226
Misc. stone and clay products	0.0020	0.0060	0.0030
Chemical industry	0.0000	0.0021	0.0011
General machinery	3.0247	5.5556	2.7778
Petroleum refinery products	0.3339	0.1982	0.1982
Repair of motor vehicles	1.2841	0.3330	0.3330
Employment[b]	0.0057	0.0032	0.0018
CO_2[c]	0.0043	0.0027	0.0027
B. Generation of shredder dust (ton per ton of EHA treated)			
TV set	0.9040	0.1517	0.1043
Refrigerator	0.5039	0.2488	0.1792
Washing machine	0.4486	0.2290	0.1386
Air conditioner	0.5256	0.1759	0.1467

[a]The unit is 10^3 yen per ton of EHA unless otherwise stated
[b]Person per ton. Includes employment for collection and transport
[c]Direct emission due to the consumption of petroleum products and lime stone in ton-Carbon

2.5 Implementing the recycling of recovered materials

2.5.1 Iron scrap

We now turn to implementation into the WIO model of the recycling of recovered materials, and start from iron components (**Table 4**). The electric steel industry operating electric arc furnaces is the largest user of iron scrap: one ton of its output requires 0.95 tons of iron scrap. On the other hand, for the steel producers operating converter furnaces, the main input is pig iron from a blast furnace (see h_{31} and h_{32} in the middle panel of Table 4), and the input of iron scrap amounts to only 0.08 tons per unit output. Provided that these input patterns remain constant, the recycling of recovered iron scraps of 0.27 million tons under **Rc** requires, with other things being equal, an increase in the output of electric arc furnace steel of 0.31 million tons (12.5 billion yen in value terms (see Δy_{32} and Δx_{32})) with a corresponding reduction in the output of converter steel (see Δy_{31} and Δx_{31}).

The major user of electric steel and converter steel is the manufacturer of hot-rolled steel. Therefore, the additional recycling of iron scrap in the amount of 0.27 million tons requires a corresponding change in the composition of electric steel and converter steel as input in the production of hot-rolled steel: the input of electric arc steel per unit of hot-rolled steel is increased by 0.00225, whereas the input of converter steel is reduced by 0.00197. Implicit in this is the assumption that both types of steel are substitutable for each other within the range of volumes considered.

2.5.2 Non-ferrous metals, plastics and glass

We next turn to the recycling of non-ferrous metal scrap, glass and plastics (**Table 5**). The use of these recycled materials as substitutes for virgin materials is a well established practice in the industry except for the material recycling of plastics. The required increase in the use of recycled materials was implemented by changing the corresponding input and waste generation coefficients. For instance, the additional use of recovered copper scraps of 0.039 million tons in the copper elongation sector is assumed to require a corresponding increase in the input coefficient for copper scraps (0.067 ton per million yen) and a decrease in the input coefficient for virgin copper (0.019 million yen per million yen) in that sector.

The additional recycling of aluminum scraps in the aluminum rolling industry, that of waste glass (glass cullet) in the glass manufacturing industry, and that of waste plastics are implemented in a similar manner by increasing the input coefficient (decreasing the net waste generation coefficient) of recovered waste materials and decreasing the input coefficient of corresponding virgin materials. Here, too, we assume, within the range under consideration, that virgin materials and the corresponding recycled materials are fully substitutable for each other.

Table 4: Implementing the Recycling of Iron Scraps in WIO

Iron scraps to be recycled	270.7	[10^3 ton]	w
Output of 'Hot rolled steel' sector	5,526	[10^9 yen]	x_{33}
Input of iron scraps per output of alternative crude steel			
'Crude steel (converters)'	0.0805	[ton/ton]	$h_{31} = p_{31} \times g_{5,31}$
	2.309	[10^{-6} ton/yen]	$g_{5,31}$
'Crude steel (electric furnaces)'	0.9469	[ton/ton]	$h_{32} = p_{32} \times g_{5,32}$
	23.760	[10^{-6} ton /yen]	$g_{5,32}$
Increase in output of crude steel that is necessary to absorb recovered iron scraps			
'Crude steel (converters)'	–312.47	[10^3 ton]	$\Delta y_{31} = w/(h_{31} - h_{32})$
	–10.900	[10^9 yen]	$\Delta x_{31} = p_{31} \times \Delta y_{31}$
'Crude steel (electric furnaces)'	312.47	[10^3 ton]	$\Delta y_{32} = -\Delta y_{31}$
	12.453	[10^9 yen]	$\Delta x_{32} = p_{32} \times \Delta y_{32}$
Incremental change in input coefficients of 'Hot rolled steel' sector			
'Crude steel (converters)'	–0.00197	[yen/yen]	$\Delta a_{31,33} = \Delta x_{31} / x_{33}$
'Crude steel (electric furnaces)'	0.00225	[yen/yen]	$\Delta a_{32,33} = \Delta x_{32} / x_{33}$

Note: The numbers occurring as suffixes in the far right column refer to sectoral classification numbers; 33 for 'hot rolled steel', 31 for 'crude steel (converters)', 32 for 'crude steel (electric furnaces)', and 5 for 'iron scraps'. p_{31} (= h_{31} / $g_{5,31}$) refers to the price of converter steel and p_{32} (= h_{32} / $g_{5,32}$) refers to the price of electric steel 10^6 yen/ton.

Table 5: Implementing the Substitution of Virgin Materials by Recovered Materials

Copper			
Copper scraps to be recycled	39.0	[10^3 ton]	w
'Rolled and drawn copper and copper alloys' sector			
Output	578.1	[10^9 yen]	x_{44}
Price of virgin copper	0.2816	[10^6 yen/ton]	p_{38}
Incremental change in input coefficients of 'Rolled and drawn copper and copper alloys' sector			
Copper scraps	0.06743	[10^{-6} ton/yen]	$\Delta g_{7,44} = w / x_{44}$
Virgin copper	–0.01899	[yen/yen]	$\Delta a_{38,44} = -p_{38} \times \Delta g_{7,44}$
Aluminum			
Aluminum scraps to be recycled	16.7	[10^3 ton]	w
'Rolled and drawn aluminum' sector			
Output	1,236	[10^9 yen]	x_{45}
Price of virgin aluminum	0.1985	[10^6 yen/ton]	p_{40}
Incremental change in input coefficients of 'Rolled and drawn aluminum' sector			
Aluminum scraps	0.01355	[10^{-6} ton /yen]	$\Delta g_{8,45} = w / x_{45}$
Virgin aluminum	–0.00269	[yen/yen]	$\Delta a_{40,45} = -p_{40} \times \Delta g_{8,45}$
Glass			
Glass cullet to be recycled	139.5	[10^3 ton]	w
'Glass products' sector			
Output	1,745	[10^9 yen]	x_{26}
Price of raw minerals (silica stone)	0.0022	[10^6 yen/ton]	p_5
Incremental change in input coefficients of 'Glass products' sector			
Glass cullet	0.07992	[10^{-6} ton /yen]	$\Delta g_{10,26} = w / x_{26}$
Raw minerals	–0.00018	[yen/yen]	$\Delta a_{5,26} = -p_5 \times \Delta g_{10,26}$
Plastics for material recycling			
Recovered plastics to be recycled			
PP	51.8	[10^3 ton]	$w_{(1)}$
PS	55.3	[10^3 ton]	$w_{(2)}$
Total	107.1	[10^3 ton]	$w = w_{(1)} + w_{(2)}$
'Plastic products' sector			
Output	10,107	[10^9 yen]	x_{23}
Price of virgin plastics			
PP	0.1420	[10^6 yen/ton]	$p_{(1)}$
PS	0.2145	[10^6 yen/ton]	$p_{(2)}$
Average	0.1794	[10^6 yen/ton]	$p_{19} = (p_{(1)} w_{(1)} + p_{(2)} w_{(2)}) / w$
Incremental change in input coefficients of 'Household electric appliances' sector			
Recovered plastics	0.01059	[10^{-6} ton /yen]	$\Delta g_{4,23} = w / x_{23}$
Virgin plastics	–0.00190	[yen/yen]	$\Delta a_{19,23} = -p_{19} \times \Delta g_{4,23}$

Note: The numbers occurring as suffixes in the far right column refer to sectoral classification numbers

2.5.3 Plastics for chemical recycling

Finally, we turn to the implementation of the injection of waste plastics into blast furnaces (**Table 6**). This case is remarkable in that it is concerned with the substitution of heterogeneous materials, plastics and coke/pulverized coal, serving the same purpose as reductant in the production of iron. It is necessary to convert them into an equivalent unit of measurement. Based on heat capacity, 1 kg of waste plastics is regarded as an equivalent of 1.31 kg of coke [19].

Because coke and plastics have different carbon content (.8856 kg-C/kg and .7288 kg-C/kg), the use of one ton of waste plastics in place of (the heat equivalent quantity of) coke can reduce the emission of CO_2 by $1.31 \times 0.8856 - 0.7288 = 0.4313$ t-C. On the other hand, waste plastics need to be pretreated into forms that are appropriate for injection into blast furnaces. This requires additional electricity of 267 kWh per ton of waste plastics. The amount of maintenance and repair corresponds to the annualized construction cost of a typical pretreatment plant characterized by 4,400 yen per ton of capacity and a durability of 10 years. As was the case for Table 3, we regard the general machinery industry as the sole supplier of the whole plant.

3 Results

Table 7 shows the major results obtained by applying the method in Section 1 to the WIO data [11] for the scenarios that were set up in Section 2. In the table, 'CO_2', besides that originating from the burning of fossil fuels and limestone, contains the GWP100 (global warming potential over 100 years) CO_2-equivalent value of methane originating from biomass fermenting at landfill sites. Inclusion of the CO_2-equivalent value of CFCs contained in EL-EHA (Table 1) [20] in this GWP100 gives 'CO_2 with CFC'. Given the modest share (less than 1%) of EL-EHA in the total volume of landfill consumption, it is not surprising that the overall 'macro effects' are rather small. Still, Table 7 provides many interesting findings.

3.1 Effectiveness of recycling

First, recycling (**Rc**), in terms of the reduction of CO_2 emission (both with and without CFCs), the volume of landfill consumption, and the demand for abiotic resources, performs better than landfilling (**Lf**) and shredding (**Sh**). Note that 'Other mining' in the classification of the WIO table consists of metallic ores, non-metallic ores except for ceramics, coal mining, crude petroleum and natural gas. Together with 'Materials for ceramics' it provides a WIO counterpart of abiotic resources. The fact that the WIO data refer only to domestic sources, as mentioned in Section 1, and that the domestic production of 'Other mining' constitutes only a small portion of the total supply, does not affect this observation, since the demand for imports is assumed to be proportional to the demand for its domestic counterparts, as in any IOA model.

While it may be obvious that increased recycling reduces landfill consumption, its effect on CO_2 emission is not obvious because the material recovery process itself is rather energy intensive. In fact, the overall demand for power (electricity) is increased by about 0.02% under **Rc** (panel **B**). On the other hand, the increase in recycling of recovered metals reduces the production of virgin metals (copper by 2.3%, pig iron by 0.4%, and aluminum by 0.4%), the production of which is quite energy intensive, and also reduces the generation of such waste as slag, waste acid, and waste alkali (panel **C**). In particular, the reduction in the demand for coal products of 0.3% can be attributed, among other things, to the reduction in the production of pig iron (due to the increase in electric arc steel)

Table 6: Implementing the Injection of Waste Plastics in Blast Furnaces

Waste plastics to be recycled (injected)	105.2	[10^3 ton]	w
'Pig iron' sector			
Output	1,197	[10^9 yen]	x_{29}
Price of coke	0.0129	[10^6 yen/ton]	p_{21}
Coke to be substituted	137.8	[10^3 ton]	$-\Delta y_{21,29} = 1.31 \times w$
Reduction of CO_2 due to the substitution of plastics for coke	45.4	[10^3 ton-C]	$-\Delta e = 0.4313 \times w$
Inputs for shredding of plastics injected			
Electricity	0.58910	[10^9 yen]	$\Delta y_{62,29} = 267\text{kWh} / \text{ton} \times w$
Machinery (Maintenance and repair)	0.46290	[10^9 yen]	$\Delta x_{52,29} = 0.0044 \times w$
Incremental change in input and emission coefficients of 'Pig iron' sector			
Coke	–0.00149	[yen/yen]	$\Delta a_{21,29} = p_{21} \times \Delta y_{21,29} / x_{29}$
Electricity	0.00049	[yen/yen]	$\Delta a_{62,29} = p_{62} \times \Delta y_{62,29} / x_{29}$
Machinery	0.00039	[yen/yen]	$\Delta a_{52,29} = \Delta x_{52,29} / x_{29}$
Waste plastics	0.08787	[10^{-6} ton/yen]	$\Delta g_{4,29} = w / x_{29}$
CO_2	–0.03790	[10^{-6} ton-C/yen]	$\Delta r_{29} = \Delta e / x_{29}$

Note: The numbers occurring as suffixes in the far right column refer to sectoral classification numbers. The price of electricity p_{62} is given in Table 2. The inventory data on $\Delta y_{21,29}$ and $\Delta y_{62,29}$ are due to [19].

Table 7: Major Results of WIO on EL-EHA

Scenarios	Recovery/Use			Extended Life	
	Shredding (Sr)	Recycling (Rc)	Recycling with DfD (DfD)	Patience (ExP)	Upgrading (ExU)
A. Macro effects					
CO_2	–0.023	–0.040	–0.041	–0.266	–0.041
CO_2 with CFC	–0.022	–1.912	–1.914	–2.509	–2.291
Incineration	0.000	–0.002	–0.005	–0.147	–0.063
Landfilling (m^3)	–0.257	–1.006	–1.135	–1.107	–1.079
Employment	0.007	0.003	0.000	–0.274	–0.015
B. Effects on industrial output					
Materials for ceramics	–0.037	–0.154	–0.154	–0.321	–0.160
Other mining	–0.019	–0.104	–0.105	–0.336	–0.122
Chemical industry	0.001	0.000	–0.094	–0.349	–0.224
Petroleum products	0.004	0.002	–0.007	–0.165	–0.011
Coal products	–0.102	–0.334	–0.202	–0.618	–0.215
Plastic products	0.001	0.001	–0.005	–1.259	–0.976
Rubber products	0.014	0.008	0.003	–0.436	0.300
Glass products	0.001	0.000	–0.006	–0.450	–0.222
Pig iron	–0.373	–0.375	–0.378	–0.770	–0.308
Crude steel (converters)	–0.451	–0.455	–0.457	–0.840	–0.396
Crude steel (electric furnaces)	0.943	0.952	0.950	0.107	0.574
Copper	0.000	–2.316	–2.318	–2.834	–2.484
Aluminum (inc. regenerated aluminum)	0.005	–0.386	–0.389	–1.127	–0.739
Rolled and drawn copper and copper alloys	0.001	0.002	0.000	–2.440	–1.944
Rolled and drawn aluminum	0.001	0.001	–0.002	–0.969	–0.712
General machinery	0.009	0.018	0.008	–0.156	1.585
Household electric appliances	0.000	0.000	0.000	–23.203	–22.542
Electric power	0.016	0.020	0.004	–0.274	0.035
Water supply	0.000	0.000	–0.004	–0.161	–0.039
Repair of motor vehicles	0.015	0.004	0.002	–0.109	0.001
Repair of machines	0.001	–0.002	–0.010	0.377	30.824
C. Effects on net waste emission					
Incineration ash	0.004	0.002	–0.015	–0.320	–0.121
Slag	–0.040	–0.063	–0.077	–2.359	–0.623
Waste oil	0.000	–0.008	–0.033	–0.574	–0.171
Waste acid	–0.001	–0.005	–0.021	–1.283	–0.935
Waste alkali	0.001	–0.047	–0.067	–1.143	–0.777
Molten slag	0.001	–0.004	–0.012	–0.383	–0.216

The numbers refer to the rate of change in percentage relative to the reference value obtained under the landfilling scenario. 'Household electrical appliances' include EHA, but other electronic and electric appliances as well.

and the use of plastics in blast furnaces. As a whole, the effect of recycling on reducing CO_2 emission turned out to be large enough to outweigh the opposite effects.

We now compare the above results with a preceding study on the recycling of EL-EHA that is based on the (conventional) method of process LCA [21]. The system under investigation in the study consisted of a recycling plant [2] and the manufacturing process of virgin materials (iron, copper, aluminum, and glass). The scenarios considered consist of **Lf**, **Sh** and **Rc**. It was found that **Rc** performs superior to **Lf** and **Sh** with regard to final waste disposal (landfilling), depletion of mineral resources, ozone layer destruction and global warming. Our results are thus fully consistent, qualitatively at least, with that of [21], in spite of the difference in the methodology in use and the definition of the system boundary.

The effect on overall employment in panel **A** will be the only item in our results that could be used as a proxy for macroeconomic effects. Note that the level of final demand in our analysis, and hence the level of GDP, is kept constant except for the extended life scenario without upgrading (**ExP**). An increase in employment then indicates a decline in the level of overall labor productivity because a larger amount of labor input is required to produce the same level of GDP. The results indicate that **Rc** will cost more labor than **Lf**, but will cost less labor than **Sh**, due largely to the efficiency of its collection system.

In order to have an idea of the importance of transport distances in these results, a sensitivity analysis was conducted by changing the one-way transport distance of collected EL-EHA under **Rc** from 12 km to 4800 km. The results in **Fig. 1** indicate that, in terms of CO_2 emission (without CFC), **Rc** scores better than **Sh** even when the transport distance is increased to 1200 km, and better than **Lf** even when it is increased to 2400 km (the full length of the Japanese archipelago is about 3000 km!). The latter would roughly correspond to an extreme

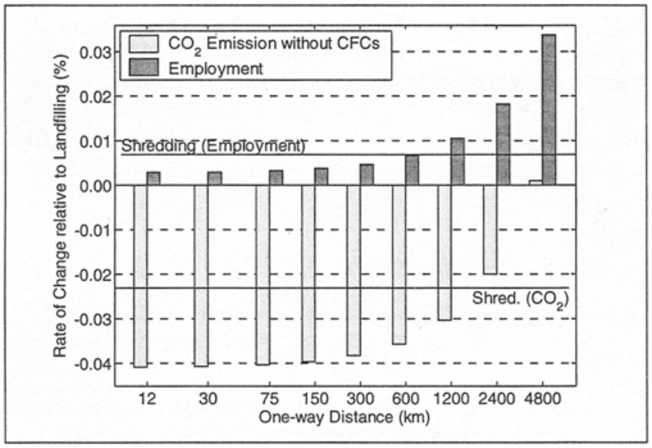

Fig. 1: Effects of transport distance on CO_2 emission and labor cost under intensive recycling

case where there is only one recycling facility in the whole country. The advantage of **Rc** with regard to CO_2 emission over **Sh** and **Lf** thus appears robust to changes in transport distance. While the labor cost tends to increase with transport distance, it does not exceed the level of **Sh** for distances under 600 km.

3.2 DfD

We next consider the effects of implementing DfD under **Rc** (**DfD**). Its implementation reduces the consumption of landfill (due to the reduced amount of shredder residue), the demand for chemical and petroleum products (due to an increase in the closed loop recycling of waste plastics), and the generation of waste such as waste oil, waste acid and waste alkali that is closely related to the chemical industry. While the reduced use of waste plastics in blast furnaces increases the demand for coal products, the overall emission of CO_2 is slightly reduced. In short, the positive environmental effects of **Rc** are augmented under **DfD**.

3.3 Extension of product life

The columns **ExP** and **ExU** refer to the extension of product life without and with the possibility of functional upgrading. Under both the scenarios, the output of 'Household electric appliances', which include EHA, decreases by 23% because of the reduction in the purchase of new products. With regard to the output level of machine repair, however, substantially different effects emerge. Under **ExU**, the reallocation of expenditure, from new purchase to repair and maintenance, leads to a compensating increase in the output of machine repair by 30%, whereas such a compensating increase in expenditure does not exist in the case of **ExP**. This produces significantly different effects between the two scenarios.

It is obvious that the level of emissions declines with a decline in the level of economic activity. The significant decline in the CO_2 emission of 0.27% (a six-fold increase compared to the recycling scenario **Rc**) under **ExP** is due to the decline in the total consumer expenditure of 0.48%. Also significant is the reduction in the demand for mineral resources ('Materials for ceramics' and 'Other mining') of 0.3% and in the amount of incineration of 0.15%. The economic cost associated with these environmental gains, however, is not small, with an almost 0.3% reduction in total employment, which amounts to 0.18 million employees! The presence of the negative impact on employment is found to be robust to alterations in the relationship between the expenditure for repair and the life extension: even for the case where the repair is increased by 500%, the rate of reduction in employment remains well above 0.2%. While 'deep ecologists' may feel comfortable with the consequence of this scenario, most people would not. A society plagued by a high unemployment rate would not be 'socially sustainable'.

Scenario **ExU** deals with an alternative case where the fall in the purchase of new EHA is compensated for by the same amount of increase in the expenditure for repair and maintenance. The large negative impact on employment that prevailed under **ExP** now disappears, while the environmental gains become smaller. Still, the environmental performance of **ExU** is significantly better than **Rc** in terms of CO_2 (with CFCs), the amount of incineration, abiotic resources, and waste generation, and is better or equally good in terms of landfill consumption and CO_2.

Recall that a reduction in resource inputs with a given level of GDP (final demand) indicates an increase in the overall productivity (efficiency) of the economy, because the same level of living standard can then be achieved with a smaller input of resources. The large decline in labor input (employment) obtained under **ExP** is no indication of an increase in productivity, because the level of GDP was not kept constant but was actually reduced. Of the scenarios where the level of GDP is kept unaltered, we observe the largest rate of decline in the overall level of employment under **ExU**. The same applies to the demand for mineral resources ('Materials for ceramics' and 'Other mining') as well. It then follows that this scenario is the most cost effective among the scenarios that keep the level of living standard unchanged. **ExU** thus emerges as the one that is sound in terms of both environmental effect and economic cost.

Our scenarios, **DfD** and **ExU**, are hypothetical and subject to considerable uncertainty. Among others, it was assumed above that the implementation of DfD and extended product life does not cause any change in the manufacturing of EHA. It is usually the case, however, that the implementation of these design concepts calls for the use of more input: upgradability calls for redundancy, and reusability calls for the use of more durable parts and components. With other things being equal, the use of more input implies a productivity decline in the manufacturing of EHA. In terms of WIO, these occur as an increase in the input coefficients of the EHA manufacturing sector.

A sensitivity analysis was carried out for alternative cases where the input coefficients of the EHA manufacturing sector are increased from 1% to 5% under both **DfD** and **ExU** (**Fig. 2**): the bars on the far right refer to the results in Table 7 where there is no decline in the productivity of EHA manufacturing. For **ExU**, the emission-saving effect was found robust to a productivity decline of up to 4%, whereas it applied to a decline of up to 3% for **DfD**. Of similar significance, we also considered a variant of **DfD** where the productivity of shredding does not increase by 50%, as above, but remains unchanged, and found that the result was negligible.

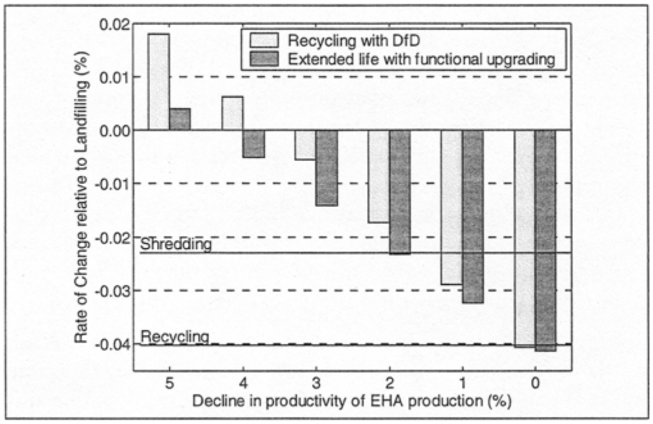

Fig. 2: Effects of decline in productivity of EHA manufacturing on CO_2 emission. The light gray bar indicates the effect on overall emission of CO_2 when the productivity of the EHA (electric home appliances) manufacturing sector is decreased by 5%, 3%, 1%, and 0% under the recycling with DfD scenario. The dark gray bar indicates the corresponding effect under extended life with the functional upgrading scenario

4 Concluding Remarks

We presented an alternative LCA methodology of EL-EHA, which is based on the WIO model. Our results indicate that the intensive recycling à la [2] is superior to landfilling and simple shredding in reducing final waste disposal, depletion of mineral resources, ozone layer destruction and global warming. It is certain that these results are conditional to a number of simplifying assumptions, some of which may not be borne out in the reality. However, their consistency with the result of another study based on a different methodology [21] indicates that the 'Japan model' of EL-EHA recycling is effective in reducing these factors of environmental impacts.

Due largely to the lack of data that are compatible with the WIO model, we were able to consider only a limited range of environmental factors. In particular, the factors concerned with human toxicity and ecotoxicity were not considered, except for a brief mention of solder recovery in the recycling process. It remains to be seen if the 'Japan model' is effective in reducing these and other factors that were omitted in this study.

An advantage of the hybrid LCA methodology presented in this article, over the conventional one based on process modeling alone, consists of its easiness in implementing a wide range of alternative scenarios with regard not only to the manufacturing phase, but also to the phase of use and end of life. This feature was demonstrated by considering scenarios with regard to the implementation of DfD and the extension of product life with and without functional upgrading. In particular, it was shown that the extension of product life could significantly reduce environmental load without having negative effects on economic activity and employment when the drop in the expenditure for new purchases is compensated for by an increased expenditure on repair and maintenance. Extension of functional product life via maintenance and upgrading constitutes an important element of ecodesign. The result thus indicates the possible effectiveness of ecodesign strategy toward the realization of a sustainable economy.

Because the introduction of ecodesign concepts in the manufacturing of appliances is still very far from widespread, related quantitative information that could be used for LCA is extremely scarce. Our results on ecodesign are thus of a limited nature conditional on hypothetical data, which need to be revised when data of better quality become available. Further research is required to provide a more accurate evaluation of ecodesign strategies, the widespread use of which could be crucial for realizing a sustainable economy.

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